

Ceriodaphnia Bioassays of Cuyahoga River Sediments, Upstream and Downstream of the Akron Water Pollution Control Station¹

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ABSTRACT. The toxicity of sediments collected from depositional and erosional substrates at two locations on the Cuyahoga River in northeastern Ohio was compared. Upstream and downstream locations from the Akron Water Pollution Control Station (Akron WPCS) were selected to evaluate effects of the Akron WPCS effluent on sediment toxicity. Seven-day *Ceriodaphnia dubia* bioassays were conducted on sediment elutriates. The no-observed effect concentrations (NOECs) were determined for survival and reproduction for each sample collected. The results indicated that sediments collected upstream from the Akron WPCS were more toxic than those collected downstream. Chlorine released with the WPCS effluent may oxidize possible contaminants resulting in lower sediment toxicity downstream. Sediments from erosional substrates were more toxic than those from depositional substrates at both locations. Erosional substrates, in contrast to depositional substrates, had small inorganic particles and contained epilithic algal/microbial mats. These factors probably influenced the concentration of contaminants.

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INTRODUCTION

The Cuyahoga River is a relatively small tributary of Lake Erie located in heavily populated and industrialized northeastern Ohio (Fig. 1). The lower Cuyahoga River, the stretch between Akron and Cleveland, is severely polluted. Sources of pollution include heavy metals, toxic organic pollutants, hazardous waste sites, agricultural activities, sewer overflows, and contaminated sediments (Willis and Foran 1986). Effluent from the Akron Water Pollution Control Station (Akron WPCS) is a major source of pollution in the lower river (Olive 1976, Trauben and Olive 1983, Willis and Foran 1986).

Based on a variety of chemical and biological water quality studies, the International Joint Commission (IJC) has designated the lower Cuyahoga River as one of 42 "Areas of Concern" in the Great Lakes Basin requiring immediate clean-up efforts (International Joint Commission 1985). The IJC has indicated that contaminated sediments are a "significant" problem in Areas of Concern and published methods recommended for the characterization of toxic substance problems in these areas (International Joint Commission 1987).

Sediments can act as both "sinks" and "sources" of toxic substances in aquatic ecosystems. Many organic and inorganic chemicals become associated with particulate matter in the water column and are incorporated in sediments when particulate matter is deposited. Contaminant concentrations in sediments can become much higher than in overlying waters (Lee and Jones 1987). Sometimes sediments can accumulate harmful levels of chemicals without overlying waters violating water quality criteria (Lyman 1987). Sediments, therefore, can be "sinks" for toxic substances.

Sediments can act as sources of toxic substances when contaminants are partitioned from sediments to the water column or to aquatic biota (Salomons et al. 1987, Adams

1987). Sediments are important components of aquatic ecosystems because they provide habitat, feeding, and breeding areas for many aquatic biota (Lyman 1987). Yet, historically they have rarely been implicated as sources of contaminants relative to discharges into surface water (Dickson and Rodgers 1985).

No single currently available method can provide enough information on the toxicity of sediments to protect aquatic ecosystems adequately (Lyman 1987, Cairns and Pratt 1989). For this reason, Chapman and Long (1983)

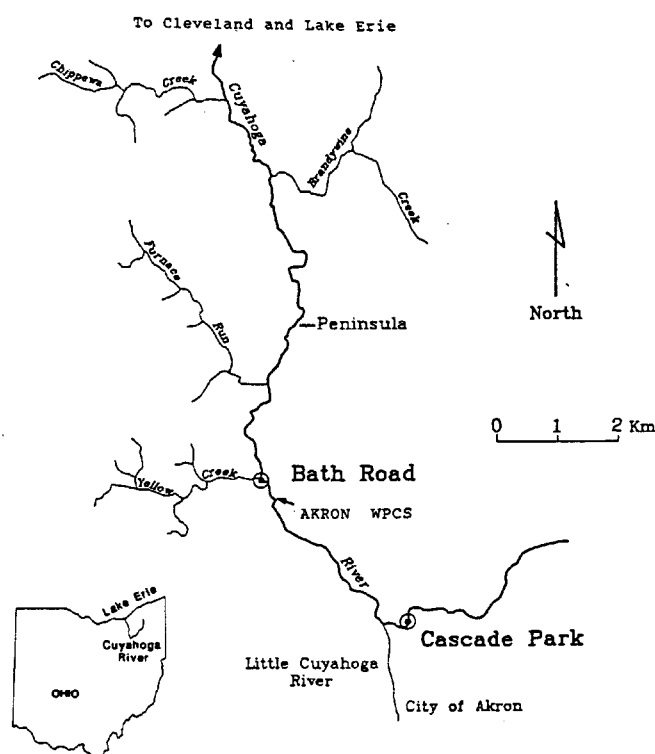


FIGURE 1. Map of the Cuyahoga River. The Akron Water Pollution Control Station (Akron WPCS), and the sampling stations at Cascade Park and Bath Road are shown.

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proposed a three component approach for sediment assessment consisting of: 1) bulk chemical analysis (concentrations of toxic chemicals); 2) functional bioassessment (toxicity bioassays); and, 3) structural bioassessment (species composition and diversity).

Chemical analyses and structural bioassessment studies were conducted in the lower Cuyahoga River near the Akron WPCS prior to this study. Twenty-three toxic organic contaminants were found in sediments upstream, while only ten were found downstream of the Akron WPCS (Willis and Foran 1986). Thus, chemical tests for the presence of sediment contaminants indicated that sediment toxicity was less acute downstream from the Akron WPCS. Field surveys of benthic macroinvertebrates, however, have indicated lower species diversity and greater numbers of pollution-tolerant species, both indicators of adverse instream effects, at locations downstream from the Akron WPCS compared to locations upstream (Olive 1976, Trauben and Olive 1983, Ohio EPA 1984).

The objective of this study was to determine the impact of the Akron WPCS on sediment toxicity. To make this assessment, 7-day *Ceriodaphnia* bioassays were conducted on elutriates from sediments collected upstream and downstream from the WPCS. The relative toxicity of sediments from erosional and depositional substrates also was compared. Presented here are the first published bioassays of sediments from this region of the Cuyahoga River.

MATERIALS AND METHODS

Sample Collection

The Akron WPCS discharges effluent into the Cuyahoga River 60.0 km from the mouth of the river at Lake Erie. Two sites that have been used by the Ohio Environmental Protection Agency (Ohio EPA) for water quality studies were selected for sediment sampling, one upstream from the Akron WPCS and one downstream (Fig. 1). These stations, referred to as Cascade Park (upstream) and Bath Road (downstream) are 68.2 km and 59.2 km, respectively, from the river mouth. Superficial sediments, less than 2 cm thick as recommended by Anderson et al. (1987), were taken from depositional substrates with a modified shovel sampler (Prater et al. 1977). Sediments from erosional substrates were obtained by scraping superficial materials from large rocks with a stiff brush. Sediment samples were stored in polyethelene bags and placed in an ice chest for transport to the laboratory where they were immediately frozen at -4°C .

All sediments were collected during the autumn of 1989. Because of changing river conditions, it was possible to collect sediments from both erosional and depositional substrates at both stations on only one sampling date, 6 November.

Elutriate Preparation

Bioassays were conducted using sediment elutriates rather than whole sediments, because the primary route of exposure for most aquatic organisms to contaminated sediments is through contact with the sediment interstitial (pore) water and not the solid sediment phase directly (Adams et al. 1985, Adams 1987). Test solutions were

prepared on the day prior to beginning a toxicity bioassay. Sediments were thawed, weighed (wet weight), and combined with dilution water to make a 600 g/l sediment:water slurry. The 600 g/l sediment solution corresponds approximately to a 1:4 volumetric sediment:water ratio. Daniels et al. (1989) recommended a 1:4 sediment:water ratio in preparing elutriates based on recent work using different elutriation techniques and sediment:water ratios.

The 600 g/l solution was stirred manually for 5 min, followed by centrifugation for 5 min at $1500 \times G$. The supernatant (elutriate) was decanted from the centrifuge tubes and pooled. Pure elutriate was used as the highest test concentration (600 g/l). Dilution water was combined with elutriate to form intermediate test concentrations of 180 g/l, 60 g/l, 18 g/l, and 6 g/l. The control was dilution water alone. Once prepared, the test and control solutions were stored at 4°C .

Dilution water (pH 8.3 and total hardness of 110 mg/l as calcium carbonate) used for culturing organisms and in bioassays was prepared by adding one part well water to four parts glass distilled water. Untreated well water was obtained from the city of Canton (OH) well field.

Toxicity Bioassays

The 7-day *Ceriodaphnia dubia* chronic toxicity bioassay (Mount and Norberg 1984) was selected to evaluate the toxicity of sediment elutriates because sediment toxicity is more likely to be chronic than acute as a result of small concentrations of contaminants released over time (Anderson et al. 1987). Test organisms were obtained from the United States Environmental Protection Agency (USEPA) Environmental Research Laboratory in Duluth, MN. Procedures for conducting bioassays have been modified from Weber et al. (1989).

The bioassay method involved placing 10 female *Ceriodaphnia*, 24 h old or younger, in individual cultures for each of 5 test concentrations (6 g/l, 18 g/l, 60 g/l, 180 g/l, and 600 g/l of sediment in culture water) and 1 control (0 g/l). Organisms were fed 0.1 ml of yeast-CEROPHYLL® trout food mixture daily. The original test organism in each individual culture container was transferred to another individual culture container with a fresh solution of the same concentration of sediment on days 3 and 5. On days 3 and 5 at the time of transfer, and on day 7 at the termination of the test, the original test organism in each culture was determined to be alive or dead and, if present, offspring were counted and excluded from further testing.

Information on survival and reproduction was then used to determine percent survival and mean number of offspring per organism per week for each sediment concentration and the control. From these data, four measures of the degree of toxicity, described below, were determined for each sample.

The no-observed effect concentrations (NOECs) based on survival (the number of organisms that are alive after 72 hours) and reproduction (the number of offspring produced per organism per 7-d) were calculated for each sediment sample collected. The NOEC is the highest test concentration that exhibits no statistically significant difference in biological response (survival or reproduction)

compared to the control. No-observed effect concentrations can be used to determine relative toxicity by indicating the amount a sediment would have to be diluted in order to be nontoxic if resuspended *in situ* (Giesy and Hoke 1989).

No-observed effect concentrations were calculated based on the procedures presented by Weber et al. (1989). Survival NOECs were calculated using Fisher's exact test. Reproduction NOECs were calculated using Dunnett's Procedure when the underlying assumptions of normality and homogeneity of variance were met. Normality and homogeneity of variance were formally tested using the Shapiro-Wilk's Test for normality, and Bartlett's Test for homogeneity of variance. If either of these tests failed, Steel's Many-one Rank Test was used to determine the NOEC based on reproduction.

The 72-hour median lethal concentration (72-h LC50), the concentration of sediment elutriate that caused a 50% reduction in survival of organisms in 72 hours as compared to control organisms, and the 7-day median inhibition concentration (7-day IC50), the concentration of sediment elutriate that caused a 50% reduction in the number of offspring produced in 7 days as compared to the number of offspring produced by control organisms, were determined. For both the LC50 and the IC50, confidence intervals can be determined, which make these endpoints useful for comparing toxicities of samples.

The 72-h LC50s and 95% confidence limits were determined using the EPA Probit Analysis Computer Program (Weber et al. 1989) when appropriate. For cases in which probit analysis is not appropriate, such as when a drastic drop in survival occurs between two successive test concentrations, the moving average-angle method of determining LC50s and associated confidence limits was used (Harris 1959). The IC50s and associated 94% confidence limits were determined using a point estimation technique and a bootstrapping procedure, respectively (Oris et al. In Press).

RESULTS

Field Observations

Visual inspection indicated that sediments from erosional substrates tended to have a greater proportion of small particles (silt and clay sized) than sediments from depositional substrates. Epilithic biological mats were associated with sediments collected from erosional substrates. It is likely that these epilithic mats were responsible for sedimentation by slowing water velocities, trapping particulate matter, and forming detritus. The dominant nonbacterial epilithic organisms were the filamentous green alga, *Oedogonium*, at Cascade Park and the diatoms, *Melosira* and *Tabellaria*, at Bath Road.

Bioassays

The NOECs based on survival and reproduction, as well as 72-h LC50s with 95% confidence limits and 7-d IC50s with 94% confidence limits for each *Ceriodaphnia dubia* bioassay were determined (Table 1).

When sediments were collected from equivalent substrates at the same sampling station on more than one date, identical NOECs were obtained. A NOEC of greater than 600 g/l indicates that no statistically significant biological

response was measured (no toxicity was detected).

The relative toxicity of sediments from the two stations can be evaluated by comparing the NOECs obtained from each substrate type independently. The relative toxicity of sediments from the two substrate types can be evaluated by comparing the NOECs obtained from each station independently.

Survival and reproduction NOECs were lower for Cascade Park sediments than for Bath Road sediments for both substrate types. Survival and reproduction NOECs were lower for sediments from erosional substrates than sediments from depositional substrates at both stations. Since lower NOECs mean higher toxicity, the results indicate that Cascade Park sediments were more toxic than Bath Road sediments, and sediments from erosional substrates were more toxic than sediments from depositional substrates.

The percent survival and mean number of young per female for each test concentration and control for bioassays of sediments collected on 6 November 1989 (the only day when both substrate types were sampled at both stations) were determined (Table 2).

The LC50s were determined for sediments collected on 6 November (Fig. 2). Sediments collected at Cascade Park from both erosional and depositional substrates had lower LC50s, and consequently were more toxic, than sediments collected at Bath Road. However, there was no significant difference in toxicity indicated by LC50s between sediments collected from erosional and depositional substrates on 6 November.

In accord with the results of the LC50s, the IC50s for sediments collected on 6 November (Fig. 3) indicate that Cascade Park sediments were significantly more toxic than Bath Road sediments for both substrate types. However, in contrast to the LC50 data, a significant difference also was indicated between sediments from erosional and depositional substrates at both sampling stations. Erosional sediments were more toxic.

DISCUSSION

Effect of the Akron WPCS

Sediments collected upstream (Cascade Park Station) from the Akron WPCS were more toxic than those downstream (Bath Road Station) (Table 1; Figs. 2, 3). This difference was expected based on tests for the presence of organic contaminants and pesticides (Willis and Foran 1986), but unexpected based on benthic invertebrate surveys (Olive 1976, Trauben and Olive 1983, Ohio EPA 1984).

The increased level of toxicity, indicated by benthic invertebrate studies, downstream from the discharge area may be caused by the release of chlorine with the Akron WPCS effluent. Giesy and Hoke (1989) point out that short term stressors (such as chlorine) can eradicate populations of benthic organisms without leaving toxic residues in the sediments. This concept might help to explain why the benthic invertebrate community is more indicative of a contaminated environment downstream of the Akron WPCS than upstream. Yet, sediment bioassays and chemical analyses indicate that sediments are less toxic downstream. Chlorine also may be able to oxidize substances that would otherwise become sediment contaminants.

TABLE 1

Results of Ceriodaphnia bioassays of Cuyahoga River sediments collected from erosional and depositional substrates at the Cascade Park and Bath Road sampling stations in 1989. No-observed effect concentrations (NOECs) and median lethal concentrations (LC50s) based on survival data, as well as NOECs and median inhibition concentrations (IC50s) based on reproduction data are shown. Confidence intervals (95% for LC50s and 94% for IC50s) are given in parentheses. All values are in g/l.

	SURVIVAL			REPRODUCTION	
	Date	NOEC	LC50	NOEC	IC50
CASCADE PARK					
Erosional	10–16	60	103.9 (65.7–164.3)	18	40.6 (17.6–50.9)
	10–30	60	67.0 (35.4–108.3)	18	41.7 (27.7–42.3)
	11–06	60	107.6 (60.7–193.2)	18	67.4 (19.2–72.4)
Depositional	11–06	180	228.7 (131.1–367.3)	180	354.1 (271.9–390.0)
	11–20	180	384.1 (236.6–651.4)	180	494.3 (451.7–538.9)
BATH ROAD					
Erosional	11–06	180	544.5 (340.0–1004.6)	60	418.9 (369.3–479.9)
Depositional	10–16	>600	>600	>600	>600
	10–30	>600	>600	>600	>600
	11–06	>600	>600	>600	>600
	11–20	>600	>600	>600	>600

TABLE 2

Percent survival and mean number of offspring (standard error) per female for each test concentration and control for Ceriodaphnia bioassays of sediments collected 6 November 1989.

Conc. (g/l)	Erosional		Depositional	
	% survival	# young	% survival	# young
CASCADE PARK				
0	100	24.6(3.45)	100	14.4(1.80)
6	80	19.4(4.28)	100	20.0(2.42)
18	90	21.5(2.75)	90	15.6(2.52)
60	80	13.6(3.56)*	100	25.4(1.95)
180	30*	3.3(1.04)*	80	16.1(3.17)
600	0*	0*	0*	0*
BATH ROAD				
0	100	27.5(0.92)	100	15.6(3.32)
6	100	27.8(0.71)	90	20.1(2.69)
18	100	27.4(0.78)	80	19.3(3.79)
60	100	27.5(1.29)	100	22.5(1.32)
180	100	22.9(1.21)*	100	19.6(2.66)
600	40*	6.9(1.27)*	90	14.7(3.06)

*indicates significantly different ($p < .05$) than control (0 g/l)

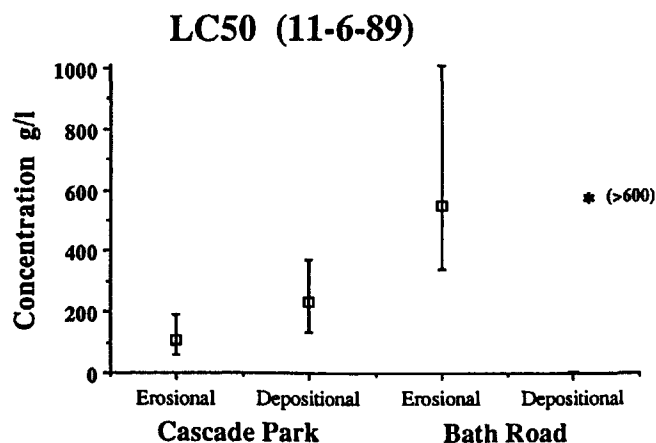


FIGURE 2. Median lethal concentrations (LC50s) for 7-day *Ceriodaphnia* bioassays on elutriates from sediments collected 6 November 1989 from erosional and depositional substrates at the Cascade Park and Bath Road sampling stations, on the Cuyahoga River. Error bars represent 95% confidence intervals. (* indicates that no toxicity was detected at the highest sediment concentration tested - 600 g/l).

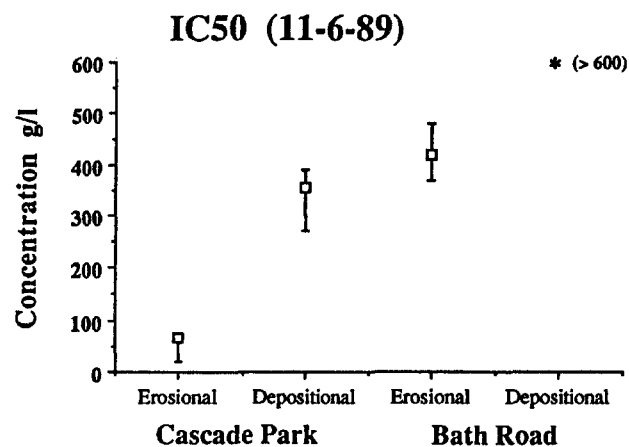


FIGURE 3. Median inhibition concentrations (IC50s) for 7-day *Ceriodaphnia* bioassays on elutriates from sediments collected 6 November 1989 from erosional and depositional substrates at the Cascade Park and Bath Road sampling stations, on the Cuyahoga River. Error bars represent 94% confidence intervals. (* indicates that no toxicity was detected at the highest sediment concentration tested – 600 g/l).

Effect of Substrate Type on Toxicity

The NOECs and IC50s indicate that sediments collected from erosional substrates were more toxic than those collected from depositional substrates (Table 1, Fig. 3). However, LC50s indicate no significant difference between the two substrate types (Fig. 2).

Since LC50s measure acute effects (short term effects resulting in death) and IC50s measure chronic effects (longer term effects resulting in a decrease in reproduction), the data suggest that there is no significant difference in acute toxicity between erosional and depositional substrates, but there is a significant difference in chronic toxicity.

Particle size may influence the toxicity of sediments. It was noted earlier that sediment samples from erosional substrates contained a greater proportion of small particles (clay and silt-sized) than sediments from depositional substrates. In bioassays of Detroit River sediments, Reynoldson and Zarull (1989) found that fine sediments were usually associated with greater levels of contamination. Because the sorption-capacities are much greater for clay than sand or silt-sized particles (Lee and Jones 1987), sediments from erosional substrates may be able to adsorb greater concentrations of contaminants than sediments from depositional substrates.

The relative organic content of sediments from erosional and depositional substrates also may affect toxicity. Sediments from erosional substrates were collected by scraping sediments from rocks. The major component in forming these epilithic sediments were biological mats. Associated with these mats were decaying organic matter and trapped particulate matter. Thus, a large portion of the sediments from erosional environments was composed of organic matter, both living and nonliving. The depositional-substrate sediments lacked comparable biological mats.

Sediment toxicity is dependent on the bioavailability of contaminants. In sediments spiked with Kepone™, Adams et al. (1985) found that acute and chronic toxicity to midges (*Chironomus tentans*) was a function of sediment organic carbon levels. In a review paper on the bioavailability of neutral lipophilic organic chemicals,

Adams (1987) concluded that organic carbon is the "primary controlling factor of the bioavailability of neutral organic chemicals sorbed to sediment." Thus, the higher organic carbon content of sediments from erosional substrates may be contributing to higher toxicity.

Other sources of toxicity also may be associated with epilithic biological mats from erosional substrates. Since 90% of all carbon in aquatic ecosystems is nonliving and much of this is in the detrital pool (Cairns and Pratt 1989), bacteria can be an important influence on sediment toxicity. "Many environmental samples have 10^6 – 10^8 bacteria per g or ml" (Trevors 1989). Breteler and Saksa (1985) found that decaying organic plant matter plays an important role in sequestering metals from solution. Contaminants sorbed by organic matter can be released by bacterial decomposition making them potentially bioavailable and toxic (Lee and Jones 1987).

Bacteria also form biofilms that are able to complex metal ions readily (Trevors 1989). There is evidence that some mercury-resistant bacterial strains can methylate inorganic mercury to methyl mercury, a form considered to be more toxic to most organisms (Trevors 1986). While bacteria can detoxify many contaminants, they also can form metabolic by-products that are toxic to other organisms.

Thus, higher organic carbon concentrations and bacteria numbers associated with epilithic biological mats may be factors contributing to toxicity of sediments from erosional substrates. If sediments from erosional substrates have the potential to be more toxic than sediments from depositional substrates, then substrate type may be an important factor to consider in hazard assessment.

Sediments obtained using recommended methods of sampling to be used in bioassays in Areas of Concern (International Joint Commission 1987) are likely to be from depositional rather than erosional environments. If erosional substrate toxicity is a significant factor in aquatic environments, methods should be included for assessing sediments from erosional as well as depositional substrates. Since a greater number of benthic invertebrates inhabit erosional substrates than depositional substrates (Hynes 1970), assessments of water and sediment quality based on benthic invertebrate surveys also should be conducted recognizing the potential of sediments from erosional substrates to be more toxic.

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LITERATURE CITED

- Adams, W. J. 1987 Bioavailability of neutral lipophilic organic chemicals contained in sediments. In: K. L. Dickson, A. W. Maki, and W. A. Brungs, eds. Fate and Effects of Sediment-Bound Chemicals in Aquatic Systems. Pergamon Press, NY. p. 219–244.
- , R. A. Kimerle, and R. G. Mosher 1985 Aquatic safety assessment of chemicals sorbed to sediments. In: R. D. Cardwell, R. Purdy, and R. C. Bahner, eds. Aquatic Toxicology and Hazard Assessment: Seventh Symposium, ASTM STP 854. Am. Soc. for Testing and Materials, Philadelphia, PA. p. 429–453.
- Anderson, J., W. Birge, J. Gentile, J. Lake, J. Rodgers, Jr., and R. Swartz 1987 Synopsis of Discussion Session 3. Biological Effects, Bioaccumulation, and Ecotoxicology of Sediment-Associated Chemicals. In: K. L. Dickson, A. W. Maki, and W. A. Brungs, eds. Fate

- and Effects of Sediment-Bound Chemicals in Aquatic Systems, Pergamon Press, NY. p. 267-296.
- Breteler, R. J. and F. I. Saksa 1985 The role of sediment organic matter on sorption-desorption reactions and bioavailability of mercury and cadmium in an intertidal ecosystem. *In*: R. D. Cardwell, R. Purdy, and R. C. Bahner, eds. Aquatic Toxicology and Hazard Assessment: Seventh Symposium, ASTM STP 854. Am. Soc. for Testing and Materials, Philadelphia, PA.
- Cairns, J., Jr. and J. R. Pratt 1989 The scientific basis of bioassays. *Hydrobiologia* 188/189: 5-20.
- Chapman, P. M. and E. R. Long 1983 The use of bioassays as a comprehensive approach to marine pollution assessment. *Mar. Pollut. Bull.* 14: 81-84.
- Daniels, S. A., M. Munawar, and C. I. Mayfield 1989 An improved elutriation technique for the bioassessment of sediment contaminants. *Hydrobiologia* 188/189: 619-631.
- Dickson, K. L. and J. H. Rodgers, Jr. 1985 Assessing the hazards of effluents in aquatic environments. *In*: H. L. Bergman, R. A. Kimerle, and A. W. Maki, eds. Environmental Hazard Assessments of Effluents, Pergamon Press, NY. p. 209-215.
- Giesy, J. P. and R. A. Hoke 1989 Freshwater sediment toxicity bioassessment: rationale for species selection and test design. *J. Great Lakes Res.* 15: 539-569.
- Harris, E. K. 1959 Confidence limits for the LD_{50} using the moving average-angle method. *Biometrics*. Sept., 1959. p. 424-432.
- Hynes, H. B. N. 1970 Ecology of Running Waters. Univ. of Toronto Press, Toronto. 555 p.
- International Joint Commission 1985 Report on Great Lakes Water Quality to the International Joint Commission, Canada and United States. Great Lakes Water Quality Board, Windsor, Ont. 212 p.
- 1987 Guidance on characterization of toxic substance problems in Areas of Concern in the Great Lakes basin. Report to the Great Lakes Water Quality Board, Windsor, Ont. 179 p.
- Lee, G. F. and R. A. Jones 1987 Water quality significance of contaminants associated with sediments: an overview. *In*: K. L. Dickson, A. W. Maki, and W. A. Brungs, eds. Fate and Effects of Sediment-Bound Chemicals in Aquatic Systems, Pergamon Press, NY. p. 3-34.
- Lyman, W. J. 1987 Establishing sediment criteria for chemicals-regulatory perspective. *In*: K. L. Dickson, A. W. Maki, and W. A. Brungs, eds. Fate and Effects of Sediment-Bound Chemicals in Aquatic Systems, Pergamon Press, NY. p. 378-387.
- Mount, D. I. and T. J. Norberg 1984 A seven-day life-cycle cladoceran toxicity test. *Environ. Toxicol. Chem.* 3: 425-434.
- Ohio EPA 1984 Unpublished benthic invertebrate data for the Cuyahoga River. Biomonitoring Section, Ohio Env. Prot. Agency. Columbus, OH.
- Olive, J. H. 1976 Chemical-physical and biological assessment of water quality in the Cuyahoga River (1973-1974). *Ohio J. Sci.* 76: 5-15.
- Prater, B. L., D. R. Barton, and J. H. Olive 1977 New sampler for shallow-water benthic macroinvertebrates. *Prog. Fish Cult.* 39: 57-58.
- Reynoldson, T. B. and M. A. Zarull 1989 The biological assessment of contaminated sediments—the Detroit River example. *Hydrobiologia* 188/189: 463-476.
- Salomons, W., N. M. de Rooij, H. Kerdijk, and J. Bril 1987 Sediments as a source for contaminants? *Hydrobiologia* 149: 13-30.
- Trauben, B. K. and J. H. Olive 1983 Benthic macroinvertebrate assessment of water quality in the Cuyahoga River, Ohio—an update. *Ohio J. Sci.* 83: 209-212.
- Trevors, J. T. 1986 Mercury methylation by bacteria. *J. Basic Microbiol.* 26: 499-504.
- 1989 The role of microbial metal resistance and detoxification mechanisms in environmental bioassay research. *Hydrobiologia* 188/189: 143-147.
- Weber, C. I., W. H. Peltier, T. J. Norberg-King, W. B. Horning, II, F. A. Kessler, J. R. Menkedick, T. W. Neihsel, P. A. Lewis, D. F. Klemm, Q. H. Pickering, E. L. Robinson, J. M. Lazorchak, L. J. Wymer, and R. W. Freyberg 1989 Short-term methods for estimating the chronic toxicity of effluents and receiving waters to freshwater organisms. U. S. Environ. Prot. Agency, Cincinnati, OH. EPA/600/ 4-89/001.
- Willis, D. P. and J. A. Foran 1986 The Cuyahoga River Study. Ohio Wildlife Federation and National Wildlife Federation. 105 p.